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Life cycle assessment of garden waste management options including long-term emissions after land application

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ABSTRACT

A life cycle assessment (LCA) was performed on five garden waste treatment practices: the production of mature compost including the woody fraction (MCIW), the production of mature compost without the woody fraction (MCWW), the production of immature compost without the woody fraction (ICWW), fresh garden waste including the woody fraction (GWIW) and fresh garden waste without the woody fraction (GWWW). The assessment included carbon sequestration after land application of the garden waste and composts, and associated emissions. The removed woody fraction was incinerated and energy recovery included as heat and electricity. The functional unit of the assessment was treatment of 1000 kg of garden waste generated in Denmark. Overall, the results showed that composting of garden waste resulted in comparable or higher environmental impact potentials (depletion of abiotic resources, marine eutrophication, and terrestrial eutrophication and acidification) than no treatment before land application. The toxicity potentials showed the highest normalised impact potentials for all the scenarios, but were unaffected by the different garden waste treatments. The choice of energy source for substituted heat and electricity production affected the performance of the different treatment scenarios with respect to climate change. The scenarios with removal of the woody fraction performed better than the scenarios without removal of the woody fraction when fossil energy sources were substituted, but performed worse when renewable energy sources were substituted. Furthermore, the study showed the importance of including long-term emission factors after land application of fresh and composted garden waste products since the greatest proportion of carbon and nitrogen emissions occurred after land application in three out of the five scenarios for carbon and in all scenarios for nitrogen.

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1. Introduction

Garden waste consists of different organic (e.g. grass and flower cuttings, hedge trimmings, tree prunings, small branches, fallen leaves, and wood debris) and inorganic (e.g. soil and stones) materials and is generated during maintenance of private gardens and public areas. Composting is one of the most common treatment options for garden waste and the resulting compost can be applied on land as a soil conditioner. Based on surveys carried out by the European Compost Network, approximately 15 million tonnes of separately collected garden waste is processed in more than 2000 European composting plants (ECN, 2018). In Denmark, garden waste roughly accounts for one fifth of municipal solid waste generated, with a *per capita* generation of about 119 kg $year^{-1}$ in 2016 (Danish Environmental Protection Agency, 2018) and the vast majority is treated in central composting plants.

During composting, the organic matter undergoes aerobic microbial decomposition resulting in generation of carbon dioxide (CO_2) and water. A common composting technology for treatment of garden waste is outdoor windrow composting where the material is laid out in elongated triangular piles, which are naturally vented. The windrows are frequently turned and the composting period can vary from weeks to months depending on the composition of the garden waste, the size of the piles and the turning frequency. After ended composting, the fresh compost undergoes further stabilisation and maturation. The product of composting is a sanitised, stable, and humus and nutrient rich compost, which can be used as soil improver, organic fertiliser, and growing medium. The use of compost can replace fossil-based products such as mineral fertilisers, peat and fossil fuels. However, greenhouse gas and ammonia (NH_3) emissions that potentially can affect the environment negatively are generated during composting (e.g.

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Andersen et al., 2010a; Boldrin et al., 2009) and a dedicated area is needed for compost production because of the extended time required for compost maturation.

There are several options for modifying the composting of garden waste to reduce environmental emissions or economic costs. First, the woody fraction, consisting of large roots, branches and trunks, can be removed from the garden waste and sent to a waste-to-energy plant. The heating value of the woody fraction is high, resulting in high energy production potential during incineration. Second, the composting period could be reduced for garden waste from which the woody fraction is removed. In this case, the remaining garden waste consists of smaller and more degradable pieces, which will compost more quickly than the larger woody material because they have a greater specific surface area available for microbial attack (Diaz et al., 2002) and because the green fraction of the garden waste is more degradable as it contains less lignocellulose and lignin than wood (Stentiford and de Bertoldi, 2011). Another option is that the compost could be land applied as immature compost after a few months of composting. Finally, garden waste could be shredded and applied directly on land, avoiding the greenhouse gas and ammonia emissions and economic costs associated with the composting process. When compost or other organic residuals are applied on land, the organic material will undergo further mineralisation resulting in a slow release of C primarily as CO_2. A part of the C will be left in the soil and the fraction left on a 100 year time frame is considered sequestered. During mineralisation, N embedded in the organic material is converted to ammonium and NH_3, and later converted to nitrate (NO_3^-) and potentially nitrous oxide (N_2O), where NO_3^- can be leached to the environment or taken up by plants and NH_3 and N_2O can be emitted to air. These effects can potentially last for many years after the application of compost or organic residual and therefore they need to be assessed in a long-term perspective. However, land application of fresh garden waste could lead to immobilisation of nitrogen (N) in the soil where crop-available inorganic N is transformed into non-available organic N, which could mean lower N availability and lead to a reduction in crop yield (Eriksen et al., 1999; Nielsen et al., 2018). Furthermore, coarsely shredded garden waste in the topsoil may also lead to difficulties with appropriate crop seedbed preparation and establishment (Guérif et al., 2001).

Reducing the amount of garden waste by taking out the woody fraction and decreasing the composting period would reduce the area needed for garden waste management and the fuel needed for compost processing. Furthermore, lower greenhouse gas and ammonia emissions are generated during the composting process and electricity and heat is produced by incineration of the woody fraction. Composting of garden waste enhances the degradation of organic matter, leading to 56–83% of organic C and 22–25% of organic N being lost to the atmosphere during composting (Boldrin et al., 2009, Komilis and Ham, 2006). When fresh garden waste or immature compost is applied on land, more N is applied, and more emissions of N containing compounds like N_2O and NO_3^- are also expected (Roy et al., 2014). Nielsen et al. (2018) conducted a simulation study to quantify long-term emissions after land application of different compost products (including fresh garden waste and immature compost) and found that matured garden waste compost exhibited a slower C turnover after land application, resulting in C sequestration factors that were 41–78% higher than the other compost and garden waste applications over a 100-year period.

In order to obtain a complete picture of the environmental profile of the various garden waste treatment options, a life cycle assessment (LCA) should be performed. Boldrin et al. (2011) performed an extensive LCA of garden waste management options and showed that increasing the proportion of garden waste that was incinerated from 6% to 35% resulted in a global warming potential saving that was three times greater with 35% incineration than with 6% incineration. Furthermore, photochemical ozone formation and acidification were not greatly affected by the chosen treatment option, while eutrophication was lowest for the scenario with home composting of approximately 18% of the garden waste (Boldrin et al., 2011). While Boldrin et al. (2011) did include peat substitution, long-term emissions associated with land application of the garden waste products were not included and the study does then not offer a complete picture of the treatment options. By long-term emissions we refer to accumulated direct emissions (in terms of NH_3, N_2O, NO_3^-) and other relevant parameters such as N-crop uptake, and C sequestered from the compost or organic residual when mixed into the soil occurring during a 100 year time frame. As reviewed by Laurent et al. (2014), LCA has been used extensively in the area of solid waste management, but the long-term effects following land application of organic waste-derived products are often not included in the assessment due to a lack of relevant input data. Yoshida et al. (2014) reviewed 35 LCA studies on sewage sludge treatment and its utilisation. Most studies (28 out of 35) included land application of sludge either accounted for as the result of avoiding the production of conventional fertilisers or the introduction of pollutants to arable land. Only 14 studies included some sort of gaseous emission, and only four considered nutrient leaching and runoff. None of the studies assessed long-term consequences of land application of sewage sludge beyond C sequestering (Yoshida et al., 2014). Simulation modelling using agroecosystem models has been used in a few studies to assess the long-term effect of land application of organic waste in life cycle assessments (Hansen et al., 2006; Turner et al., 2016; Yoshida et al., 2018). However, there is a lack of such studies on garden waste composting options that include long-term emissions after land application.

The aim of this study is to compare the environmental performance of alternative garden waste management options where the woody fraction of the garden waste is removed before composting and the composting time is reduced before land application of garden waste products. The study is of relevance to waste management companies as they are interested in optimising their operation from an environmental and economic point of view. Due to the high heating value of woody material, this fraction is a relevant input material to waste incinerators. Furthermore, the waste incinerators might have unused incineration capacity especially when considering that a larger part of the waste in the future will undergo increased source separation and material recovery, which is why incineration of the woody material of garden waste is of interest.

2. Methodology

This study followed the LCA methodology defined in ISO 14040 (2006). The description of the goal and scope and the life cycle inventory (LCI) are presented in the methodology section, while the life cycle impact assessment (LCIA) and interpretation of results can be found in the results, discussion and conclusions sections. Supplementary Information (SI) is provided online and contains detailed descriptions of assumptions, parameter values and data sources.

2.1. Goal and scope definition

The goal of this study was to assess and compare the environmental profile of:

- Mature compost including the woody fraction (MCIW) – production and land application of screened mature compost made of shredded garden waste including the main share of the woody fraction and without stones or foreign items, with a composting and maturing period of 14 months. Stones, foreign items and a small fraction of the wood were incinerated.
- Mature compost without the woody fraction (MCWW) – production and land application of screened mature compost made of shredded garden waste without the woody fraction, stones or foreign items, with a composting and maturing period of 14 months. Stones, foreign items and the whole woody fraction were incinerated.
- Immature compost without the woody fraction (ICWW) – production and land application of screened immature compost made of shredded garden waste without the woody fraction, stones or foreign items, with a composting period of three months. Stones, foreign items and the whole woody fraction were incinerated.
- Garden waste including the woody fraction (GWIW) – production and land application of fresh shredded and screened garden waste including the main share of the woody fraction. Stones, foreign items and a small fraction of the wood were incinerated.
- Garden waste without the woody fraction (GWWW) – production and land application of fresh shredded and screened garden waste without the woody fraction, stones or foreign items. Stones, foreign items and the whole woody fraction were incinerated.

These five treatment scenarios were chosen for reasons of data availability and because they represent current practice (MCIW) and feasible changes to current practice (MCWW, ICWW, GWIW and GWWW) in Denmark. The current study provides decision support at a level that does not result in large scale consequences *i.e.* at the micro-level, which corresponds to the goal Situation A as defined in the ILCD standard (ILCD, 2010). For this reason, the life cycle is modelled by depicting the existing supply-chain, *i.e.* using an attributional approach. The foreground system is modelled using specific primary data including technology-specific, long-term emissions after land application of the waste products. Impacts associated with the background system were evaluated through system expansion for upstream processes (electricity, diesel, treatment chemicals), and substitution of avoided processes was included for downstream processes (electricity, district heating, crop production and mineral P fertiliser production and use). More details are presented in SI-1 in the SI.

The functional unit of the assessment was treatment of 1000 kg of garden waste generated in Denmark. A representative garden waste composition for Denmark that includes seasonal variations can be found in Boldrin and Christensen (2010). The garden waste was assumed to consist of 24.0% woody fraction (trunks, branches and roots), 75.6% soft fraction (small branches, leaves and grass), and 0.4% stones and foreign items in wet weight (Boldrin and Christensen, 2010). The chemical composition of the garden waste is presented in SI-2 in the SI.

The system boundaries of this study included all the processes taking place after the garden waste arrived at the composting facility up to its final application on agricultural land or incineration in a waste-to-energy plant, including all emissions to air, water and soil (Fig. 1). Emissions and resource consumption were analysed up to 100 years after field application of the garden waste products. The reason for choosing 100 years in this study was the intention to use a single timeframe for soil processes and for soil C sequestration. The timeframe of 100 years is often applied for assessment of effects from greenhouse gases (Hauschild et al., 2013). The geographical scope was Denmark.

2.2. Life cycle inventory

Process modelling and sensitivity analysis were conducted using EASETECH (version 2.3.6, February 2017), a mass flow-based life cycle assessment (LCA) modelling tool (Clavreul et al., 2014). Background processes were taken from the standard EASETECH, Ecoinvent version 3.1 and ELCD databases.

Emissions and operational data for garden waste treatment were taken from a series of literature studies that were based on measurements at the Affaldscenter Aarhus in 2007 and 2008 (Andersen et al., 2010a, 2010b; Boldrin and Christensen, 2010). The first step in the pre-treatment of the garden waste was screening, removing 100% of foreign items, 100% of stones and 12.9% of total wood (large tree trunks) (*cf.* SI-3 in the SI). In the scenarios in which the woody fraction was removed (MCWW, ICWW, and GWWW), it was assumed that 100% of branches, roots and tree trunks were removed in the screening process. After screening, the garden waste was shredded. Based on Andersen et al. (2010a), it was assumed that 1.5 L diesel per tonne wet-weight garden waste was needed for pre-treatment (wood removal, screening and shredding), irrespective of the treatment.

Composting was assumed to take place in outdoor windrows that were turned approximately every six to eight weeks (Boldrin and Christensen, 2010). Estimates of emissions of CO_2, CH_4, carbon monoxide (CO) and N_2O from composting piles were based on measurements in Andersen et al. (2010a, 2010b). Emissions of NH_3 were not included in the studies by Andersen et al. (2010a, 2010b) and were therefore based on a garden waste composting study performed by Hellebrand (1998). Table 1 offers an overview of the accumulated emissions within the first three months (immature compost), and within the first 14 months for composting with and without the woody fraction (mature compost). Diesel consumption during 14 months of composting was assumed to be 1.5 L of diesel per tonne wet weight (Andersen et al., 2010a). Emissions associated with diesel consumption were included in the analysis. It was assumed that diesel consumption was evenly distributed throughout the composting process, meaning that 0.3 L of diesel per tonne wet weight was needed for three months' composting. It was assumed that no additional diesel was needed for treatment in the scenarios with land application of fresh garden waste. Electricity use at the composting plant, e.g. for illumination and office buildings, was also included, based on Andersen et al. (2010a) at 0.2 kWh per tonne wet-weight garden waste for mature compost. Electricity consumption was also assumed to be evenly distributed throughout the composting period, leading to 0.04 kWh per tonne wet-weight for immature compost. For fresh garden waste, electricity consumption was assumed to be negligible. More information about the composting process can be found in SI-4 in the SI.

After the composting process, the material was screened and particles larger than 25 mm sent for incineration. The part sent for incineration was assumed to contain 2.9% of the soft fraction, 14.1% of branches and roots and 18.9% of tree trunks (with the latter two only applicable to mature compost including the woody fraction) (Andersen et al., 2010a). The energy recovered from the combustion of waste was utilised for combined heat and power generation, with a recovery efficiency of 22% for electricity and 73% for district heating. Bottom ash was landfilled. More information about the incinerator is presented in SI-5 in the SI.

The fate of C and N after land application of garden waste products has been modelled by Nielsen et al. (2018) using the Daisy model (Hansen et al., 2012), while the fate of P is based on simulations with the phosphorus life cycle inventory (PLCI) model developed by ten Hoeve et al. (2018) (see Table 2). The Daisy model is a mechanistic and deterministic model that simulates crop growth and movement and transformation of C and N in the soil-plant-

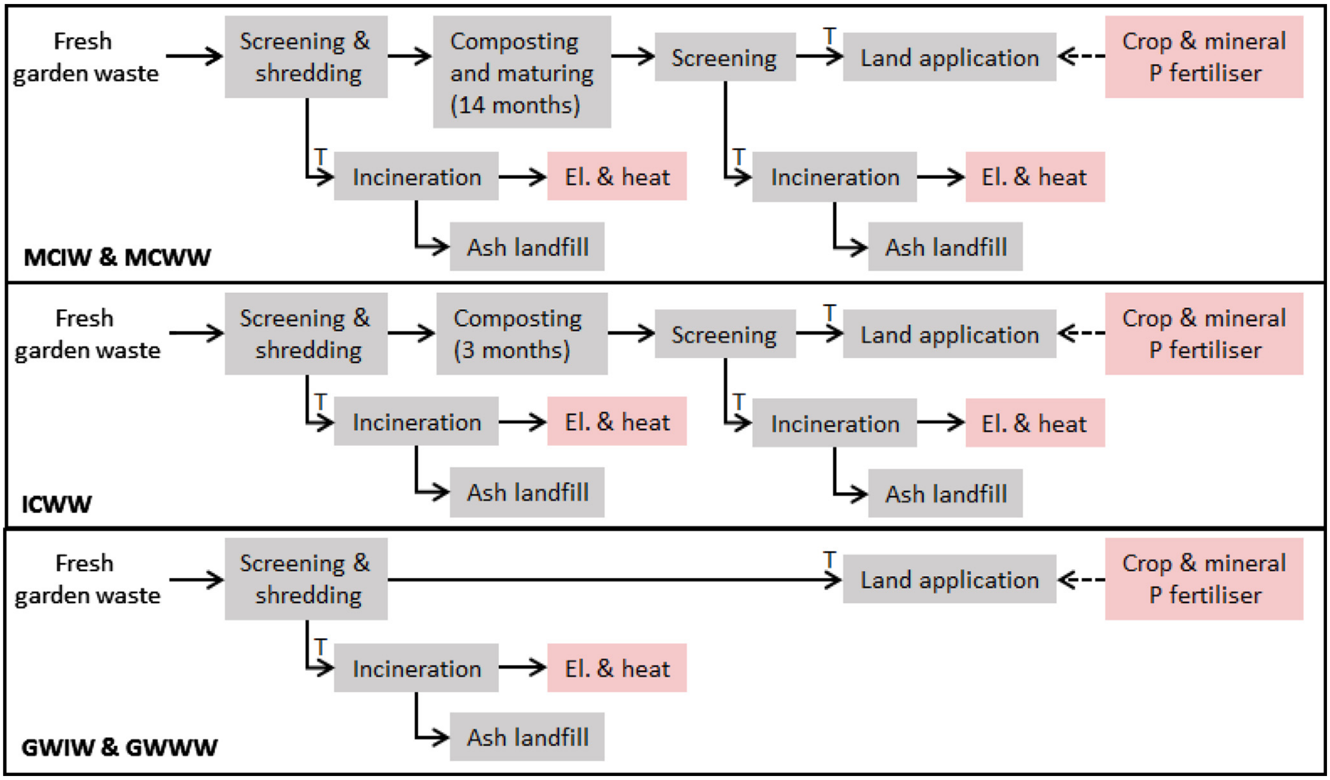

Fig. 1. System boundaries of the five sludge management options: MCIW (mature compost including the woody fraction), MCWW (mature compost without the woody fraction), ICWW (immature compost without the woody fraction), GWIW (fresh garden waste including the woody fraction), GWWW (fresh garden waste without the woody fraction). El. is electricity. T is transportation.

Table 1
Proportion of garden waste carbon (C) and nitrogen (N) lost during the composting and maturation process.

Compound	14 months of composting		3 months of composting
	Mature compost including woody fraction	Mature compost without woody fraction	Immature compost without woody fraction
CO_2-C	54.6%	72.8%	15.6%
CH_4-C	1.2%	1.6%	0.3%
CO-C	0.2%	0.2%	0.05%
NH_3-N	4.4%	4.4%	1.0%
N_2O-N	1.8%	1.8%	0.4%

atmosphere system (Hansen et al., 2012). The model has been used extensively for estimation of field emissions associated with the application of waste on land (e.g. Bruun et al., 2006, 2016; Yoshida et al., 2015). A detailed description of the model set-up can be found in SI-6 in the SI. It was assumed that the garden waste products were applied to a sandy loam soil with a medium precipitation regime for European conditions. Furthermore, high crop response conditions were assumed, *i.e.* conditions that appear when N is the limiting factor for crop growth (Yoshida et al., 2016). This can be due to low N-application and/or a low N status in the soil. Compost and other organic waste products are generally applied to agricultural fields on which no animal manure is applied. Primarily inorganic N fertilisers are used and at least in Denmark this means that N-application is relatively restricted. Restricted N-application leads to a crop response to the application of waste by taking up more N. This is what is defined as high crop response conditions by Nielsen et al. (2018). High crop response conditions also lead to low NO_3^- leaching and low additional leaching in response to the addition of N. With respect to P, the soil status can be expected to be relatively low because waste products are most likely to be applied on these soils, leading to low P losses and a high increment in crop yields in response to the addition of P

Table 2
Fate of carbon (C), nitrogen (N) and phosphorus (P) after application of garden waste products on a sandy loam soil under an average Danish precipitation regime at the fertiliser application rate within the high response phase. For further information on C and N emission factors, see Nielsen et al. (2018) and section SI-6 in SI.

Fate of C (% of C input in the respective products)		
Garden waste product	Soil C sequestration	CO_2-C emission
MCIW	25.8%	74.2%
MCWW	25.8%	74.2%
ICWW	15.1%	84.9%
GWIW	17.8%	82.2%
GWWW	14.5%	85.5%

Fate of N (% of N input in the respective products)					
Garden waste product	NH_3-N volatilisation	N_2O-N emission	NO_3^--N to groundwater	NO_3^--N to surface	N crop uptake
MCIW	0.11%	2.5%	30.6%	15.8%	17.6%
MCWW	0.11%	2.5%	30.6%	15.8%	17.6%
ICWW	0.13%	3.5%	31.0%	15.0%	15.2%
GWIW	0.18%	4.1%	15.0%	5.8%	10.6%
GWWW	0.08%	3.9%	22.4%	10.6%	17.7%

Fate of P (% of P input in the respective products)		
	Mineral P fertiliser substitution	P loss
All garden waste products	26.5%	0.32%

Note: MCIW is mature compost including the woody fraction, MCWW is mature compost without the woody fraction, ICWW is immature compost without the woody fraction, GWIW is fresh garden waste including the woody fraction, GWWW is fresh garden waste without the woody fraction. Expressed for P is the fraction of the initial application of 30 kg P ha^{-1} in garden waste products, which resulted in lower application of mineral P fertiliser (substitution, F_{PFS}) and increased P loss (F_{Ploss}) in the subsequent 100 years.

and high fertiliser substitution. Furthermore the assumption was made that all metals contained in the garden waste products were introduced to agricultural soil, but did not reach concentrations that would negatively affect crop yields. This is a realistic scenario as it is assumed that the garden waste products are land applied on agricultural soil on which no animal manure is applied.

For biogenic C in the garden waste products, between 14.5% and 25.8% was sequestered in the sandy loam soil in Denmark after 100 years, as shown by Nielsen et al. (2018) (Table 2). The sequestered C was considered a saving in greenhouse gas emission, leading to a negative climate change potential (Christensen et al., 2009). The remaining C applied on land was emitted as biogenic CO_2. Within the first days of land application, 15% of the ammoniacal N (NH_3 and NH_4^+) in the garden waste products was assumed to be volatilised as NH_3 (Nielsen et al., 2018). As ammoniacal N only constituted a very small fraction of total N in fresh and composted garden waste, emission factors expressed as a share of total N were small for NH_3, ranging from 0.08% for GWWW to 0.18% for GWIW (Table 2). The emission factors for N_2O were highest for fresh garden waste products (around 4%) and lowest for mature compost (around 2.5%) (Table 2). Nitrate (NO_3^-) leaching and runoff factors were highest for mature compost (MCIW and MCWW) (30.6% and 15.8%, respectively) and lowest for fresh garden waste including the woody fraction (GWIW) (15.0% and 5.8%, respectively) (Table 2).

Over the 100-year period after land application of the garden waste products, the net N crop uptake simulated by Nielsen et al. (2018), and correspondingly the net crop yield, was higher than without application of garden waste products. This effect did not show up directly after application for most products. For fresh garden waste that included the woody fraction, net immobilisation of N occurred during the first five years and it took 39 years to obtain a positive cumulative yield. For GWWW it took seven years until the cumulative crop N uptake was greater than zero, whereas it took three years for ICWW. Mature compost with and without the woody fraction showed a positive effect on crop yields from the year of application. Interestingly, the cumulative crop yield after 100 years for garden waste without the woody fraction was greater than for mature compost.

The phosphorus life cycle inventory (PLCI) model was used to estimate phosphorus (P) loss and substitution of mineral fertiliser (*cf.* SI-6 in the SI for an explanation of the model and how it was set up to calculate inventory data for the current scenarios). The same P loss and substitution estimates were used for all scenarios. In short, the simulations express the consequences of an initial application of garden waste product followed by ordinary fertiliser management throughout the modelling period, compared with no initial application of fertiliser P but subsequently with ordinary fertiliser management. The difference in mineral P fertiliser application (following normal P fertilisation based on soil tests and recommended values) and P loss was calculated and expressed as a proportion of the initial application of the garden waste product. To simulate P availability of the garden waste product in the soil, a partitioning coefficient to the soil labile P pool of 65% was used based on data from Möller et al. (2018). The simulations showed that the greatest share of P applied was stored in the soil (45%) or taken up by the crops (28%). Only a small fraction – approximately 0.3% – was lost to the aquatic environment. Approximately 8 kg less mineral fertiliser was applied in the model run with an initial application of garden waste product, compared with the model run without the initial application. The emission, uptake, sequestration and replacement factors applied in this study for C, N and P can be found in SI-6 in the SI. Triple superphosphate (TSP) was the mineral P fertiliser that was assumed to be replaced. TSP or monocalcium phosphate is a widely used fertiliser and the most common fertiliser, which is not associated with N (Kongshaug et al., 2014).

Nitrogen in garden and park waste applied to agricultural land does not need to be accounted for in farm fertilisation planning (N budgets) required by the Danish fertilisation ordinance (Ministry of Environment and Food of Denmark, 2012). In other words, farmers are allowed to apply as much garden waste per hectare as they wish and do not have to reduce the amount of mineral N fertiliser accordingly. However, the assumption was made in this study that the amount of compost applied corresponded to 30 kg P ha^{-1}, in accordance with the regulations on biowaste and biosolids of the Ministry of Environment and Food of Denmark (2006). Additional application of N in the form of fresh garden waste or compost leads to higher crop yields, as described above. The higher cumulative crop yields for all garden waste products were accounted for by replacement of crop production elsewhere in the world. A detailed description of avoided crop production can be found in SI-7 in the SI.

The garden waste products were assumed to be applied to agricultural land 100 km away from the composting facility. The products were transported by a 10-tonne truck and field application was performed by a tractor equipped with a solid waste spreader. The woody fraction was assumed to be incinerated at a waste incinerator located 30 km away from the composting facility.

The construction and demolition phases of the facilities were included in this study, as previous studies suggest that the contribution made by composting and incineration facilities to the environmental impacts of composting is significant (e.g. Brogaard and Christensen, 2016). More information about the inclusion of capital goods in this study can be found in SI-8 in the SI.

2.3. Life cycle impact assessment

In the life cycle impact assessment (LCIA), midpoint impact potentials were calculated for 13 impact categories: human toxicity carcinogenic (HTc), human toxicity non-carcinogenic (HTnc), eco-toxicity (ET), depletion of abiotic resources (mineral, fossil and renewable, RD), marine eutrophication potential (ME), climate change potential (CC), photochemical oxidant formation potential (POF), terrestrial eutrophication potential (TE), terrestrial acidification potential (TA), particulate matter formation potential (PM), freshwater eutrophication potential (FE), ionising radiation potential (IR) and stratospheric ozone depletion potential (SOD). For climate change potential, biogenic CO_2 emissions were not contributing while C sequestration of biogenic C was. The characterised results of each impact category were normalised by dividing the results by the annual environmental impact of an average person, which brings all the results on the same scale. As defined in the ISO standard, normalisation is a process to calculate the magnitude of the impacts, relative to a reference (ISO 14044, 2006). The impacts of an average European person were chosen as a reference. This means that the normalised impacts express the impacts of a functional unit relative to the impacts of an average European person in units of person equivalents (PE). The choice of LCIA method for each impact category was based on the recommendations in the ILCD, as presented in Hauschild et al. (2013). Normalisation references represent the EU27 for the reference year 2010 and were taken from Benini et al. (2014) and Sala et al. (2015). Table SI-9-1 in SI shows the characterisation methods and normalisation factors used for each impact category.

2.4. Sensitivity analysis

A sensitivity analysis was conducted at three levels. First, a contribution analysis was performed in order to identify the processes that have the greatest influence on the overall impact potentials. Second, for processes that showed a high contribution in the contribution analysis, the effect of changes to the most influential parameters and choices was analysed. These were the NH_3 emis-

sion factor from composting, the source of electricity and heat that was replaced by incineration of the woody fraction, and the choice of crop that was replaced elsewhere when crop production was increased as a consequence of the application of garden waste product. The NH_3 emission factor was increased and decreased by 10%, the source of electricity was altered from coal to wind power, the source of heat was altered from natural gas to wood, and the replacement of crop production was altered from production in a place with high environmental impacts, represented by the EcoInvent process for wheat production in Switzerland, to production in a place with low environmental impacts, represented by the EcoInvent process for wheat production in Germany. Third, the effect of climatic conditions and soil type was assessed by including three precipitation regimes (low: 563 mm yr^{-1}; medium: 605 mm yr^{-1}; high: 828 mm yr^{-1}) and three soil types (coarse sandy soil, sandy loam soil and clay soil), resulting in nine different combinations. Emission factors for these combinations were taken from Nielsen et al. (2018).

3. Results

3.1. Fate of carbon and nitrogen

The fate of garden waste biogenic carbon (C) and nitrogen (N) over the 100-year period is shown in Fig. 2. When looking at C, it is evident that composting and incineration reduced the amount of C that was land applied (Fig. 2a). In GWIW, the largest share of C in the initial garden waste was land applied, which led to the highest CO_2 emissions after land application of all scenarios, but also the greatest C sequestration in the soil. In the MCWW scenario less than 15% of the initial C in garden waste was actually land applied. Composting for 14 months led to the emission of 44% of initial C for MCWW and 52% for MCIW. Incineration of the woody fraction (MCWW) led to the emission of approximately 42% of the initial C, whereas this was only 10% for MCIW (Fig. 2a).

When looking at garden waste N, the greatest share ended up being land applied, ranging from 72% of N in the initial garden waste for MCWW to 92% for GWIW (Fig. 2b). For soil storage, the same pattern emerged as for C, with GWIW showing the greatest storage. Crop N uptake was lowest (10%) in the GWIW scenario of all scenarios, even though the largest amount of N was land applied here. This shows that the N in fresh garden waste where the woody fraction is still included is not available for the crops, but is bound to the soil as organic N and will result in further immobilisation of soil mineral N, also leading to the lowest NO_3^- leaching of all scenarios. Crop N uptake was highest in the MCIW scenario, with 15% of the N initially present in the garden waste. The emission of N_2 and reactive N, *i.e.* NH_3 and N_2O, from composting was small, ranging from 1% of N in the initial garden waste for ICWW to 6% in MCIW. Incineration led to the emission of approximately 22% of the initial garden waste N as N_2 in the scenarios in which the woody fraction was incinerated.

3.2. Life cycle impact assessment

The results presented in the following section are shown in Fig. 3 and in Fig. SI-10-1 to SI-10-6 in the SI. Fig. 3 shows the normalised impact potentials in person equivalents (PE) for the eight impact categories (human toxicity carcinogenic, human toxicity non-carcinogenic, ecotoxicity, abiotic resource depletion, marine eutrophication, climate change, terrestrial eutrophication and terrestrial acidification) with the highest impact potential for the five scenarios. Fig. SI-10-1 shows the characterised impact potentials for these eight impact categories, while the figures in the SI (SI-10-2 to 10-6) present normalised impact potentials for the remaining impact categories (photochemical oxidant formation, particulate matter, freshwater eutrophication, ionising radiation and stratospheric ozone depletion). The contribution analysis for all impact categories for the five scenarios can be found in Figs. SI-11-1 to SI-11-5 in the SI.

3.2.1. Toxicity

The land application stage was the main contributor to the toxicity-related impact categories of human toxicity carcinogenic (HTc), non-carcinogenic (HTnc) and ecotoxicity (ET) (Fig. 3a–c). The substitution of mineral fertiliser led to a small amount of avoided emissions contributing to toxicity, however this was very small compared to the toxicity impact from the application of heavy metals contained in the compost applied to agricultural land. Only a few metals were responsible for the main share of impacts, namely chromium (93–95% for HTc, 24% for ET), zinc (88% for HTnc, 57–58% for ET), copper (16–17% for ET), lead (1% for HTc, 8% for HTnc) and mercury (1% for HTc, 3% for HTnc). The toxicity-related impact potentials were comparable for all scenarios, where only removal of the woody fraction had a small influence on the total toxic impact potentials. Removal of the woody fraction led to lower toxicity potentials as smaller amounts of metals were land applied, but they ended up in a landfill after incineration. The main share of chromium in the garden waste products originated from the grass and leaf fraction, which constituted more than 75% of the garden waste. As it was assumed that the garden waste products would be applied to agricultural land with low soil N and P status due to minimal application of organic fertilisers in the past, it was reasonable to assume that soil copper and zinc levels were also low. If the toxicological effect of copper and zinc were not included in the analysis, HTnc would decrease by 88% and ET by 74% compared to when copper and zinc were included. Other implications of metal application to soil are included in the discussion section.

3.2.2. Depletion of abiotic resources

Construction materials for capital goods were the main contributor to depletion of abiotic resources (RD) for all scenarios, from 88% for GWIW to 99% for the composting scenarios (Fig. 3d). The RD was highest for the scenarios that included composting of the garden waste because of the production of construction materials used for the composting facility, including steel and concrete and the steel hall. The two composting scenarios where the woody fraction was incinerated showed the highest RD potential due to the need for construction materials for composting and incineration (mostly caused by reserve-based resources such as indium, nickel, cadmium and lead). Brogaard and Christensen (2016) also showed that capital goods contributed almost 100% of the resource depletion for composting of garden and park waste. Fertiliser substitution and crop substitution contributed to environmental impact savings, especially due to the substitution of the use of abiotic reserve-based resources. For the GWIW scenario, these savings were so great that it resulted in a net impact saving, meaning that fewer abiotic resources were used for garden waste management than would have been used under alternative production using mineral P fertiliser and additional wheat produced elsewhere in the world, owing to lower yields. In the GWIW scenario, only foreign items, stones and a small fraction of the wood were incinerated, meaning only a small amount of materials were needed for the construction of the incineration plant. In general, the composting facilities showed a greater RD potential than the incineration facilities.

3.2.3. Eutrophication

With respect to eutrophication, predominantly marine and terrestrial systems were affected due to N-related emissions, mainly

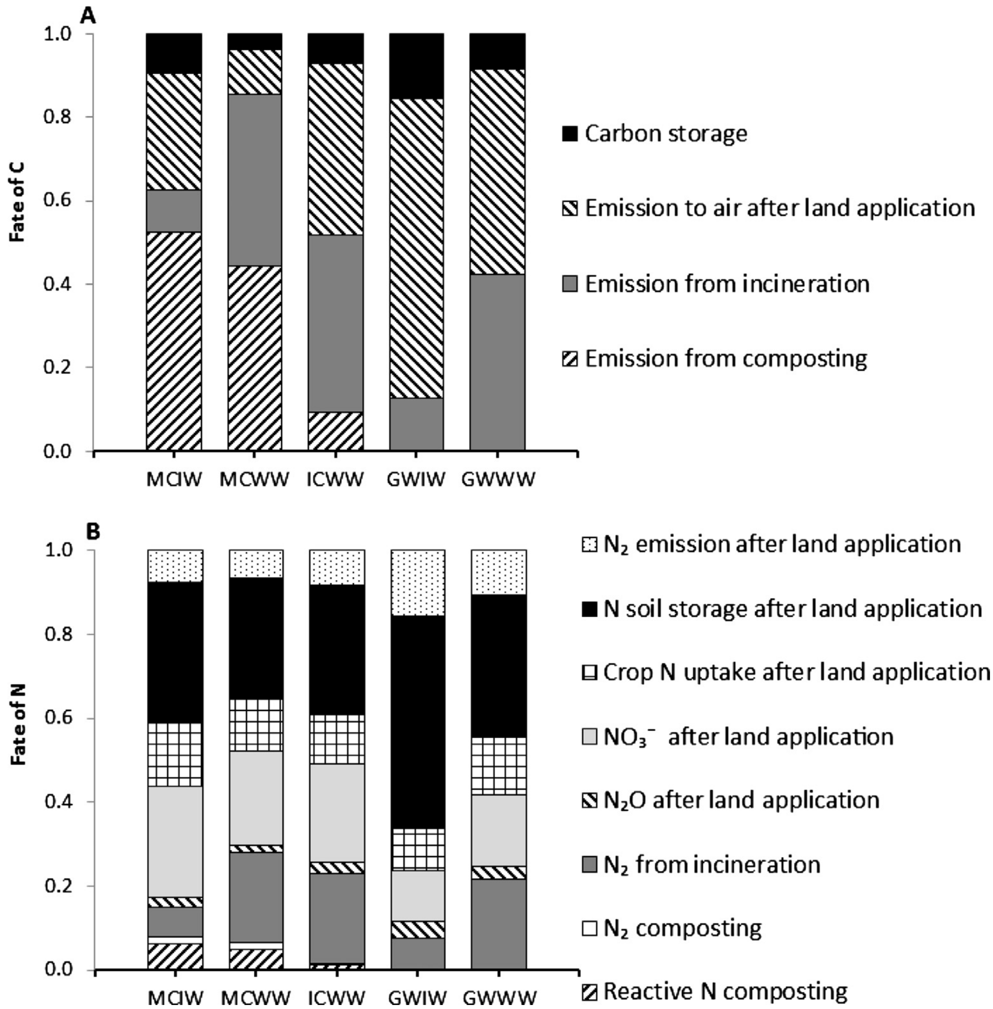

Fig. 2. Fate of biogenic carbon (A) and nitrogen (B) in garden waste in the five management options: mature compost including the woody fraction (MCIW), mature compost without the woody fraction (MCWW), immature compost without the woody fraction (ICWW), fresh garden waste including the woody fraction (GWIW) and fresh garden waste without the woody fraction (GWWW). NH_3 emissions after land application are not included in the figure as they were negligible.

of NH_3, nitrogen oxides (NO_x), and NO_3^-. Freshwater eutrophication, caused by P emissions to the environment, was negative for all the scenarios due to savings from mineral P fertiliser production and avoided crop production elsewhere in the world (Fig. SI-10-4 in the SI).

All scenarios showed a net positive marine eutrophication (ME) potential that was mainly caused by NO_3^- leaching after land application of the garden waste products (Fig. 3e). The GWIW scenario showed the lowest ME potential with 0.017 PE (corresponding to 0.28 kg N-eq in characterised impact potential), due to the lowest NO_3^- leaching of all scenarios as a result of prolonged soil N immobilisation caused by the woody fraction of the GWIW. However, this also slightly reduced crop yields for several decades after application (Fig. SI-7-1). Soil application of compost (MCIW, MCWW and ICWW) showed the highest ME with 0.036–0.039 PE (0.61–0.66 kg N-eq). Incineration of foreign items, stones and wood, and bottom ash landfill also resulted in a positive contribution to ME mainly due to emissions of NO_x, especially in the scenarios where the entire woody fraction was incinerated (MCWW and GWWW). Avoided crop production elsewhere in the world resulted in the greatest ME potential saving as NO_3^- leaching was avoided. Avoided energy production also contributed to environmental savings, mainly caused by the avoidance of NO_x emissions.

The terrestrial eutrophication (TE) potential ranged from 0.003 PE (0.6 mol N-eq) for GWIW to 0.016 PE (2.8 mol N-eq) for MCWW (Fig. 3g). The main emissions contributing to TE were NH_3 and NO_x. The main contributors to this impact category were the composting process during which NH_3 volatilises, and NO_x emissions from incineration and bottom ash landfilling. In GWIW there was no composting and only a small fraction of the waste was incinerated, while in MCWW the garden waste was composted for 14 months and the whole woody fraction was incinerated. Savings in TE were mainly caused by energy substitution for the scenarios with incineration of the woody fraction at −0.005 PE (−0.9 mol N-eq). Substitution of wheat production elsewhere in the world also contributed to TE savings for all scenarios.

3.2.4. Climate change

The climate change (CC) potential showed values ranging from −0.015 PE (−134 CO_2-eq), for the GWWW scenario to 0.005 PE (42 CO_2-eq), for the MCIW scenario (Fig. 3f). The main positive contributors to CC were CH_4 and N_2O emissions from the composting process, N_2O emissions from land application and fossil CO_2 emissions from bottom ash landfilling. However, for all scenarios apart from MCIW, soil C sequestration and substituted processes showed greater environmental savings than the positive contributions to CC. Energy substitution in particular resulted in large CC savings due to the replacement of electricity and heat production in the coal-fired combined heat and power plant. These savings were greater than the positive contributions to CC from the waste-to-

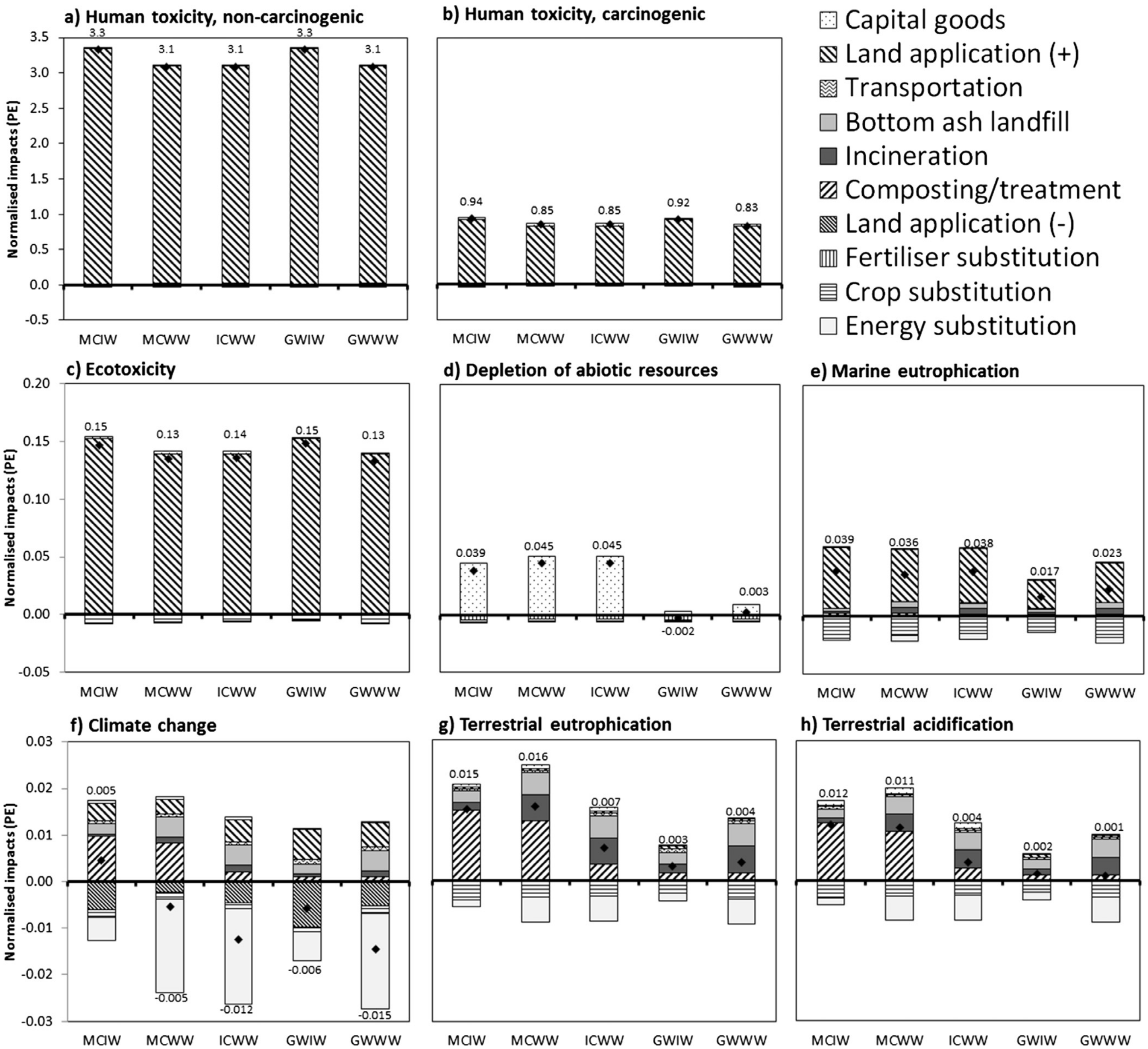

Fig. 3. Normalised impact potentials of the five garden waste treatment options: mature compost including the woody fraction (MCIW), mature compost without the woody fraction (MCWW), immature compost without the woody fraction (ICWW), fresh garden waste including the woody fraction (GWIW) and fresh garden waste without the woody fraction (GWWW) and contribution from different life cycle stages. Land application (−) is soil carbon storage, while land application (+) is the emission of N_2O.

energy plant and from fuel consumption associated with bottom ash landfilling. Soil C sequestration resulted in greater savings than the positive contribution to CC from land application for the scenarios in which the woody fraction was included in the land-applied waste (MCIW, GWIW). For the scenarios with immature compost (ICWW) and fresh garden waste without the woody fraction (GWWW), positive contributors to CC from land application and C sequestration cancelled each other out. Only in the MCWW scenario was C sequestration lower than the positive contributions from land application. Substitution of crop production elsewhere in the world led to a small environmental saving in all scenarios.

3.2.5. Other impact categories

The terrestrial acidification (TA) potential ranged from 0.001 PE (0.047 mol H^+-eq) for GWWW to 0.012 PE (0.57 mol H^+-eq) for MCIW (Fig. 3h). The main contributors to this impact category were the composting process, incineration, bottom ash landfilling and capital goods. The main emissions contributing to TA were NH_3, NO_x and SO_2. Environmental savings were caused by the substitution of crop production elsewhere in the world and heat and electricity production. The savings in TA from replacement of energy production were smaller than the positive contributions from the waste-to-energy plant and landfilling of bottom ash.

3.3. Sensitivity analysis

The contribution analysis showed that the life cycle stages that were the main contributors depended on the impact category being focused on (see Fig. SI-11-1 to SI-11-5). When looking at the overall picture, composting, land application, capital goods and energy and crop substitution were the main contributors for at least two of the impact categories.

The impact categories mainly influenced by adjustment of the NH_3 emission of the composting process in the sensitivity analysis

were TE and TA (Fig. 4). The scenarios with mature compost showed the greatest response to the alteration in NH_3 emissions, ranging from a 7.0% increase and decrease in TE for MCWW to a 9.3% increase and decrease in TA for MCIW. This is represented by the error bars in Fig. 4.

The CC impact potential in particular was affected by the assumption of the replaced source of electricity and heat (Fig. 5). In general, all scenarios showed higher impact potentials when the electricity and heat substituted were produced from renewable sources rather than from fossil energy sources. This reflects the fact that more energy was replaced than used in all scenarios apart from MCIW. The scenarios with incineration of the whole woody fraction performed better than the scenarios where most of the woody fraction was land-applied when fossil energy sources were chosen, and performed worse when renewable energy sources were chosen. This highlighted the importance of the assumption of energy source on the final results as incineration of the woody fraction changed from being the preferred treatment to the disfavoured treatment with respect to CC.

When the application of garden waste products affected the crop yields, system expansion was used to assess the effect by modelling the replacement of crop production in other parts of the world. This had a substantial effect on eutrophication (Fig. 6) The amounts of replaced crops were relatively small, *i.e.* the increase in crop production due to the use of garden waste products was relatively small, but still relatively large savings in eutrophication potential were accomplished. This was due to the fact that the replaced crops were assumed to be cultivated somewhere with high environmental impacts, represented by the EcoInvent process for wheat production in Switzerland. When the assumption was made that the replaced crops were grown in a place with low environmental impacts (represented by the EcoInvent process for wheat production in Germany), the net impact potentials increased. In other words, the environmental savings that were gained because of the increased crop production were smaller when a less burdensome crop production method was replaced. When the produced crops replaced less burdensome crop production the ME potential increased by 29% for ICWW and by 57% for GWWW, but the order of the scenarios remained unchanged (Fig. 6a). The TE potential increased by 15% for MCWW to 66% for GWWW, but the order of scenarios remained unchanged (Fig. 6b). The CC potential increased by approximately 0.001 PE (9 kg CO_2-eq) for all scenarios and the order of scenarios remained unchanged (Fig. 6c). For FE, net environmental savings changed to net environmental burdens when crops were replaced that were cultivated with low environmental burdens (Fig. 6d). This was

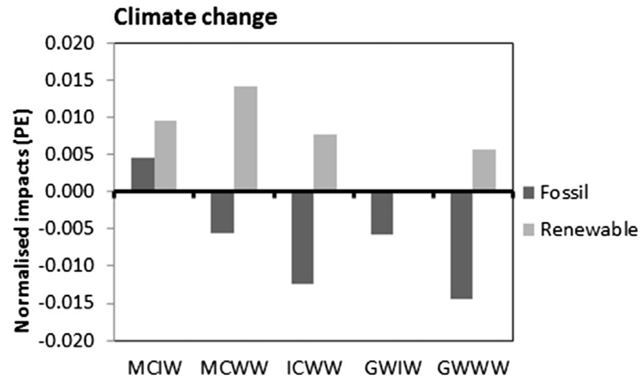

Fig. 5. Sensitivity analysis for the choice of electricity and heating source, and the effect on climate change. Dark grey bars represent the net climate change potential with coal as a source for electricity and natural gas as a source for heating, while the light grey bars represent the net climate change potential with wind power as a source for electricity and wood as a source for heating.

due to the fact that the P losses associated with the substituted crop produced in a place with low environmental burdens was much smaller than those associated with substituted production in a place with high environmental burdens. In fact, the environmental savings were greater than the environmental impacts when the replaced crop was grown in a place with high burdens, while the environmental savings were smaller than the environmental impacts when the replaced crop was grown in a place with low burdens. The order of scenarios was affected by the assumption about which crop was replaced as a consequence of increased yields: GWIW performed worse than MCIW and MCWW when the replaced crop was grown with high environmental burdens, but performed better when grown with low environmental burdens.

Fig. 7 presents the results of the analysis of the different climate and soil combinations tested in the sensitivity analysis in terms of impact potentials in ME and CC for the five garden waste scenarios. The GWIW scenario showed the greatest variation in ME depending on soil-precipitation combinations, with a difference of 0.040 PE (0.3 kg N-eq) corresponding to 465% between the lowest and highest value, and the MCWW showed the smallest variation with a difference of 0.018 PE (0.7 kg N-eq) corresponding to 152% between the lowest and highest value (Fig. 7a). For ME, the scenarios with land application of fresh garden waste generally performed better than the scenarios with land application of composted garden waste, regardless of the soil type and precipitation regime. GWWW performed best for all precipitation regimes on a coarse sandy soil, while GWIW performed best for all precipitation regimes on a sandy loam and clay soil. ICWW performed worst for all soil types and precipitation regimes, apart from medium (Denmark) and high (The Netherlands) precipitation on a sandy loam soil and high precipitation on a clay soil. In these cases, MCIW performed worst. With respect to ME, the combination of a coarse sandy soil and high precipitation regime showed the highest impact potential for all scenarios. This was due to the relatively high NO_3^- leaching caused by the low water-holding capacity of the coarse sandy soil and high precipitation causing fast percolation of water through the profile. The average ME potentials of sandy loam soils of the three precipitation regimes was lower for all scenarios than the average impact potentials of the coarse sandy and clay soils.

In general, CC was less affected by changes in climatic conditions and soil types in comparison to ME. The main contributor to CC from the land application phase was soil C sequestration, a parameter that was barely affected by soil type and climatic conditions. GWIW showed the greatest variation in CC with a difference

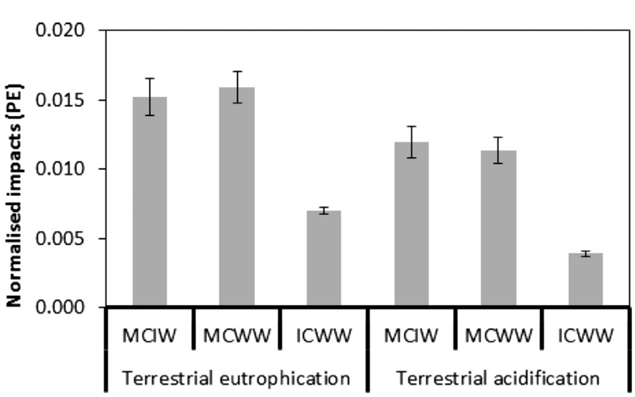

Fig. 4. Sensitivity analysis for net terrestrial eutrophication and net terrestrial acidification potential caused by increasing and decreasing the NH_3 emission from composting by 10% for the scenarios including composting. The bar represents the net impact potential, and positive and negative error bars represent the variation caused by increased and decreased emission factors respectively.

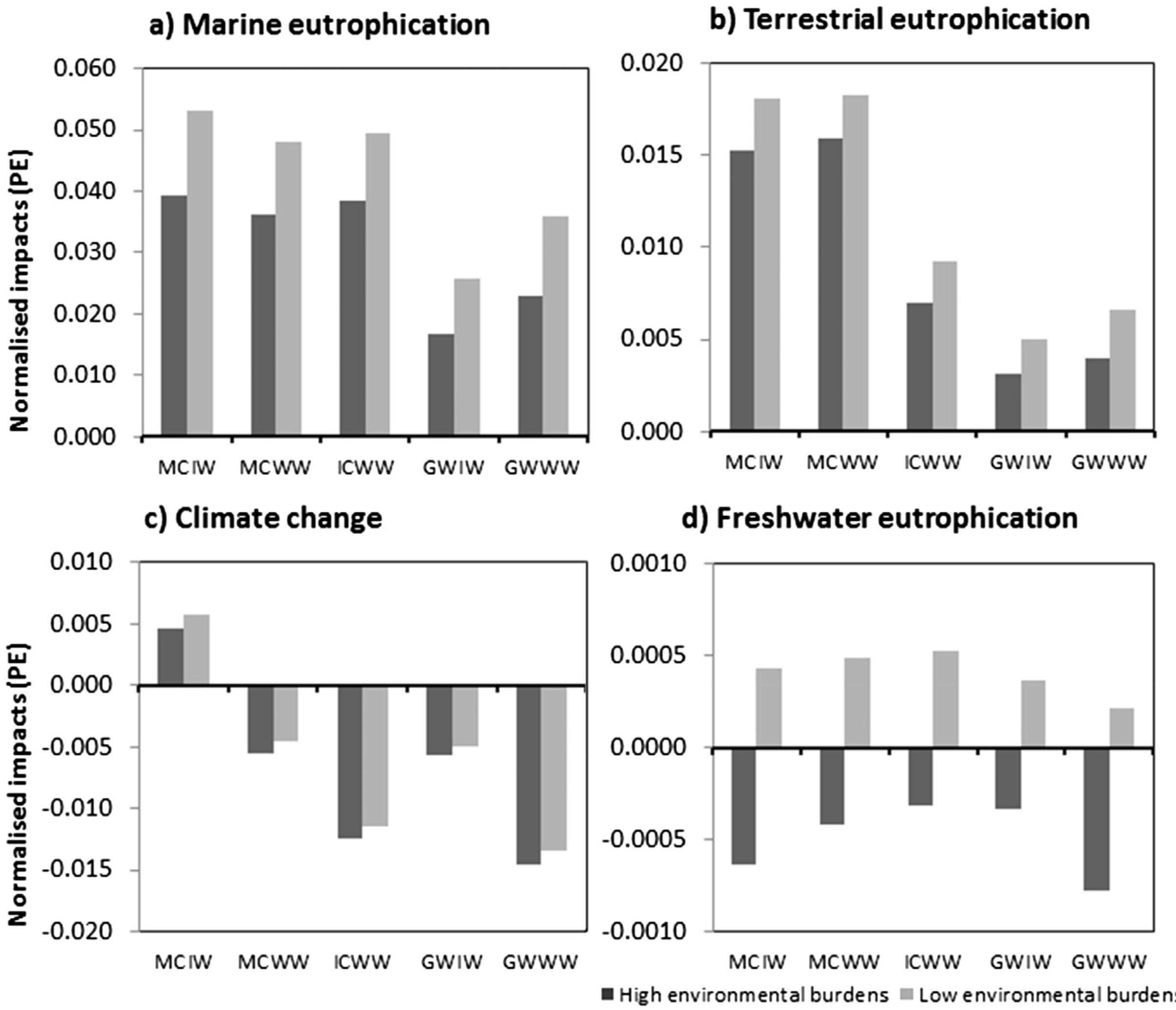

Fig. 6. Sensitivity analysis for the choice of substituted crop production and the effect on net marine eutrophication, net terrestrial eutrophication, net climate change and net freshwater eutrophication. Dark grey bars represent substituted crop production in a place with high environmental impacts, *i.e.* production in Switzerland, while the light grey bars represent substituted crop production in a place with low environmental impacts, *i.e.* production in Germany.

of 0.0063 PE (58 kg CO_2-eq) between the lowest and highest value, and MCWW showed the smallest variation with a difference of 0.0027 PE (25 kg CO_2-eq) between the lowest and highest value (Fig. 7b). For CC, the MCIW scenario always performed worst and the GWWW scenario always performed best, regardless of the soil type and precipitation regime. This was due to the high greenhouse gas emissions from the composting process in MCIW and the large greenhouse gas savings from substituted energy production in GWWW.

4. Discussion

In general, the results showed that composting of garden waste resulted in comparable or higher environmental impact potentials than no treatment before land application. For depletion of abiotic resources, marine eutrophication, and terrestrial eutrophication and acidification in particular, the fresh garden waste scenarios showed lower impact potentials than the scenarios with composting. Even when the NH_3 emissions from the composting process decreased by 10%, the composting scenarios still showed higher terrestrial eutrophication and acidification potentials than the fresh garden waste scenarios. When production of the substituted crop was changed to one with lower environmental burdens, the composting scenarios still performed more poorly than the fresh garden waste scenarios for marine eutrophication and terrestrial eutrophication and acidification.

The relative performance for marine eutrophication (ME) potential of the scenarios depended on the soil type and precipitation conditions. The soil types and precipitation conditions were representative for three European countries, namely Germany (low), Denmark (medium) and The Netherlands (high). For all combinations of soil type and precipitation conditions, scenarios with fresh garden waste performed best, while the scenario with immature compost without the woody fraction and mature compost including the woody fraction performed worst. The precipitation conditions mainly affected the magnitude of ME, on a coarse sandy soil, higher impact potentials showed for more wet conditions than for more dry conditions, while on a clay soil lower impact potentials showed for wet conditions than for more dry conditions. On a coarse sandy soil within each precipitation regime, the composting and fresh garden waste scenarios performed comparably, while the fresh garden waste scenarios performed better than the composting scenarios for sandy loam and clay soil types. From a marine eutrophication perspective, these effects of soil type highlight a possible potential for application of fresh garden waste on a sandy loam and clay soil regardless the precipitation conditions.

Theoretically, shredded fresh garden waste can be land applied without further treatment. This could be problematic because of

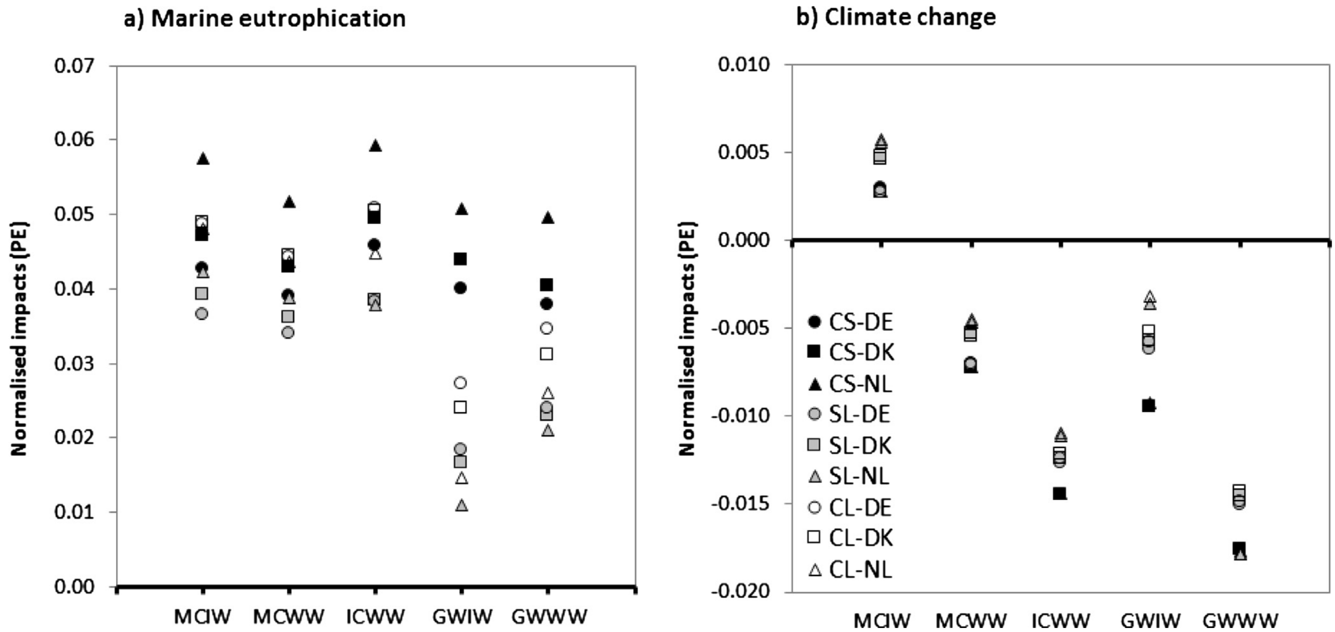

Fig. 7. Variation in net marine eutrophication potential (MEP) and net climate change potential (CCP) caused by soil and precipitation combinations. The grey squares represent the values presented earlier in the results section. Abbreviations: soil types: CS: coarse sandy soil, SL: sandy loam soil, CL: clay soil; climatic zones: DK: Denmark, medium precipitation of 605 mm yr^{-1}, DE: Germany, low precipitation of 563 mm yr^{-1}, NL: The Netherlands, high precipitation of 828 mm yr^{-1}.

immobilisation of N in the first years after application of the fresh garden waste, especially when the woody fraction is not removed (Nielsen et al., 2018). The model simulations in this study (Nielsen et al., 2018, and Fig. SI-7-1 in the SI) showed that this immobilisation will lead to a reduction of crop yields in the first years after application. It takes almost 40 years before the cumulative crop yield becomes higher when fresh garden waste including the woody fraction is land applied than when it is not (Fig. SI-7-1). From a farmer's perspective, this is a major drawback to applying fresh garden waste, as for a farmer short-term crop yields are more important than long-term effects. Farmers may also experience other agronomic implications of using either composted or freshly shredded garden waste products, which are not covered by the current assessment. The negative effects of applying coarsely shredded garden waste could be short-term problems with topsoil structure and production of phytotoxic compounds, both of which could impair proper crop establishment and have potentially significant negative crop yield impacts. Positive long-term effects from the garden waste products not covered in the current analysis include better soil fertility, water-holding capacity and tilth as a consequence of increased content of organic matter (Lal, 2006; Peltre et al., 2016).

Removal of the woody fraction from mature compost and fresh garden waste resulted in a positive effect on climate change due to fossil energy substitution after incineration of the woody fraction. The choice of energy source for heat and electricity production that was substituted was decisive here: when renewable energy sources were substituted, the scenarios with wood removal performed worse than the scenarios without wood removal. This demonstrates the importance of the choice of energy sources that are applied in the study, both for energy use and substitution. It might be reasonable to assume replacement of coal and natural gas as marginal energy sources now, but in future the marginal sources will presumably change to being renewable in origin as the Danish government strives for a fossil energy-free society by 2050 (Danish Government, 2011). The kind of renewable energy it will replace in the future is also an important question. Incineration of the woody fraction is a flexible technology that can be used when there is a need for electricity, as opposed to wind energy, for example, which can only be produced when it is windy.

Alternative garden waste management practices, *i.e.* moving away from composting to land application of fresh garden waste, show a shift in the timing of C and N emissions from waste treatment to after land application. For C, soil C sequestration and CO_2 emission from land application were greatest for the scenario with land application of fresh garden waste including the woody fraction. This scenario also showed the highest N-related emissions and storage after land application of all scenarios.

Toxicity potentials were practically unaffected by the garden waste treatment. Only removal of the woody fraction, and consequently removal of a share of the metals, led to slightly lower toxicity potentials. However, the toxicity potentials should be interpreted with caution as the literature shows that characterisation factors provided by USEtox are overestimated. Plouffe et al. (2016) present characterisation factors for zinc that are more than 50 times lower than the values applied by USEtox. This may also be the case for other heavy metals. Furthermore, it should be noted that the heavy metal contents of the garden waste products were under the threshold values set in Denmark's strict biowaste and biosolids regulations (Ministry of Environment and Food of Denmark, 2006). The concentrations of zinc and copper were more than a factor of 30 lower than the maximum applicable threshold while the concentration of chromium was more than a factor of three lower (Ministry of Environment and Food of Denmark, 2010). Another complication is that zinc and copper are essential crop nutrients, which means that for some soils they can be beneficial for plants, and therefore toxicity of the application clearly depends on the specific sites on which they are applied. USEtox only considers the amount of heavy metals added, not the effective concentrations. However, these metals only carry a toxicological risk after application on agricultural land when they are already present in high concentrations.

As mentioned in the introduction, the LCA study by Boldrin et al. (2011) showed that incineration of a larger share of the

woody fraction led to a climate change saving that was a factor of three greater than when the woody fraction was composted with the rest of the garden waste. This result is in line with the present study's finding that mature compost without the woody fraction also showed a greater saving than compost including the woody fraction with respect to climate change. However, in the present study the net climate change potential for mature compost including the woody fraction was positive because the assumption was made that the compost was applied to arable land substituting mineral P fertiliser, while Boldrin et al. (2011) made the assumption that compost was applied in private gardens, substituting peat as a soil amendment. The climate change impacts associated with peat production are much greater than those associated with mineral P fertiliser production. Furthermore, other impacts such as eutrophication associated with land application of garden waste and compost products were included in the present study, while they were omitted from the study by Boldrin et al. (2011).

There are very few LCA studies looking at different management options for garden waste. Morris et al. (2013) reviewed 82 studies of end of life options for organic waste. They compared composting, anaerobic digestion, gasification, combustion, incineration with energy recovery, mechanical biological treatment, incineration without energy recovery, and landfill disposal and found aerobic composting and anaerobic digestion to be the best options. Incineration of the woody fraction exclusively as in our MCWW, ICWW and GWWW scenarios holds the advantage that the most energy rich fraction is used for energy recovery while the nutrient rich and less energy rich part is recycled through composting. Apart from the large variability in methodology and system boundaries in the reviewed studies, this may be an important factor explaining the generally good performance of incineration in the current study.

In the scenario with removal of the woody fraction before composting to maturity (MCWW), due to a lack of data, the same emission factors during composting and after land application were applied as for the scenario with composting without removal of the woody fraction (MCIW). This simplification may be problematic, especially with respect to NH_3 volatilisation. When the woody fraction is removed, immobilisation may be reduced as the woody fraction has a high C/N ratio and therefore high immobilisation potential. This may lead to a higher concentration of ammonium during composting of the MCWW and potentially lead to higher NH_3 emissions. However, the pH of the composting material may decrease by removal of the woody fraction if the remaining, more easily degradable garden waste decomposed more quickly, leading to fermentative activity. A drop in pH will reduce the NH_3 loss. The emission factors for N_2 and N_2O during composting could also be influenced by the removal of the woody fraction. As mentioned above, mineralisation of N may increase through the removal of the woody fraction, which potentially leads to higher N_2 and N_2O emissions from the composting process. However, the recommended C/N ratio for composting is between 25 and 35. The ratio was 35 before composting of the garden waste without the woody fraction (Chowdhury et al., 2014), therefore it is not possible to identify whether emissions of N_2 and N_2O are significantly influenced or not. In order to validate the assumption that removal of the woody fraction did not influence the emission factors for N during composting, actual measurements need to be performed.

5. Conclusions and perspectives

The alternatives to conventional composting of garden waste, *i.e.* removing the woody fraction and shortening or skipping the composting process, showed environmental benefits for one or more impact categories. Climate change in particular was influenced positively by the alternative practices, as an environmental impact was changed into environmental savings when emissions from the composting process were reduced and instead energy generated from wood incineration was substituting fossil fuels. The scenarios with land application of fresh garden waste in particular performed better than the composting scenarios with respect to depletion of abiotic resources, marine eutrophication and terrestrial eutrophication and acidification. However, current agricultural practice does not include land application of fresh garden waste, presumably due to the immobilising effect of garden waste on N and related crop yield drops in the first years after application. Other effects not covered by the life cycle assessment, such as problems with establishing a good seedbed and phytotoxic effects, may also mean that direct application of garden waste is unacceptable. The results were very sensitive to the electricity and heat mix used, whether fossil or renewable energy was used, and the chosen soil and climate conditions. This study shows the importance of including long-term emission factors after land application of fresh and composted garden waste products since most of the emissions occurred after land application in almost all the scenarios for carbon and in all scenarios for nitrogen. Furthermore, the importance of considering the soil type and precipitation regime in the relative performance of the garden waste treatment scenarios was highlighted. In future LCA studies with land application of organic waste products, long-term emission factors need to be modelled based on soil type and precipitation conditions.

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Appendix A. Supplementary material

Supplementary data to this article can be found online at https://doi.org/10.1016/j.wasman.2019.01.005.

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